Preliminary assessment of water quality requirements of seagrasses in Western Port

A report prepared for Melbourne Water, December 2013

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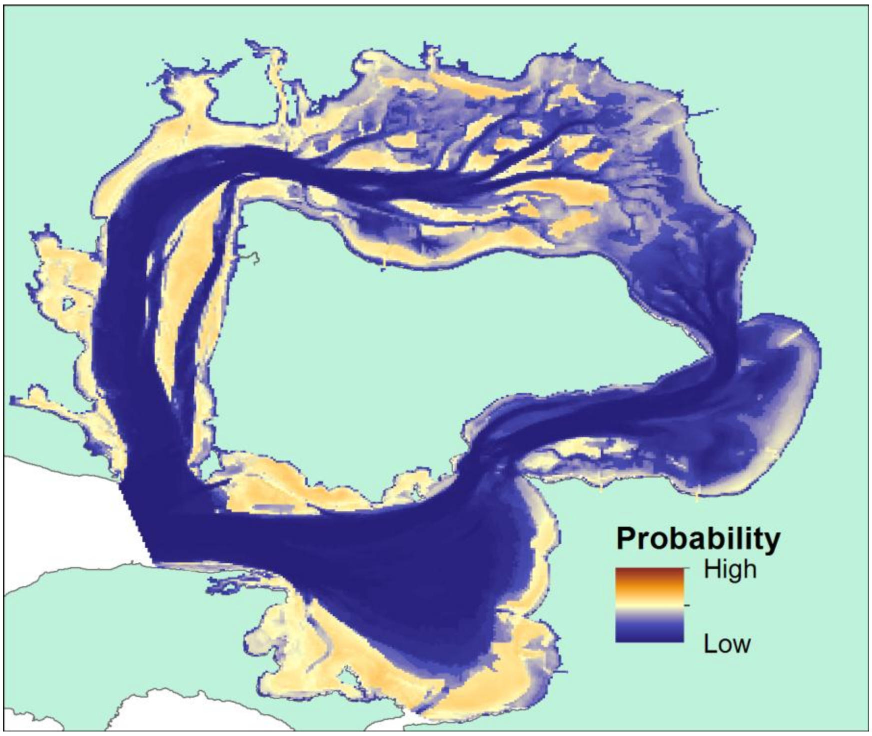

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Executive Summary

Seagrass beds are recognized as important habitats for a diverse range of fish species at various stages of their life cycles. Western Port has lost a third of its seagrass cover since the late 1970s and although there has been a recent partial recovery, many formerly vegetated areas remain bare. The aim of this project was to identify the factors that control the present day seagrass distribution in order to identify the water quality requirements for seagrass growth. Our approach was to combine previous maps of seagrass distribution, measurements of water quality and numerical models of hydrodynamics, wave characteristics and sediment transport to overlay patterns of possible controlling variables and the present distribution of seagrass to identify critical thresholds. The modelling was used to simulate three yearlong periods: 1974, 1998 and 2009 as well as a shorter calibration simulation from 2012. Highest nutrient concentrations were in the far north-west of Western Port at Watsons Inlet, where seagrass density is high. There was very little change in present-day nutrient concentrations compared to the 1970s for the entire bay. This led us to conclude that eutrophication is unlikely to be a controlling factor in the current distribution of seagrass within Western Port.

Stable isotope analyses for carbon strongly suggested seagrasses are light limited in relatively shallow water compared to neighbouring Port Phillip Bay. The maximum entropy model, Maxent, was used to model seagrass distributions from depth data and numerical model output including Total Suspended Solids (TSS), % light at the sediment surface, % time above threshold for sediment erosion/resuspension, mean wave height and mean wave period. The vegetation map from 1999 and the 1998 numerical modelling results were used to build the Maxent model, which was used to estimate seagrass and macroalgal distributions for 2011 and 1974. The model reproduced past and present distributions of seagrass well, and this tool offers a promising approach to simulate the likely distribution of seagrass based on different management scenarios for Western Port.

Light was a poor predictor of seagrass distribution because most seagrass is found in the intertidal zone, where mean light is not a good indicator of 'useful' light (light available while submerged). We found that TSS was the variable that could best explain seagrass distribution. TSS was a stronger predictor of seagrass distribution than light which suggests that TSS plays a role in seagrass distribution by smothering. Another possibility is increased seabed heights and longer inter-tidal air exposure times, but our modelling cannot account for this, and further investigation is required to test this hypothesis.

Analysis of the seagrass distribution in 1999 as a function of depth and mean modelled TSS showed thresholds of 0.007, 0.012 and 0.019 mg L^{-1} TSS for the presence of dense, medium and sparse seagrass respectively in the subtidal zone. In the intertidal, the threshold for the presence of seagrass was about 0.01 mg L^{-1}. These thresholds are, however, no guarantee of seagrass presence. These apparent TSS thresholds require further evaluation, including an understanding of the catchment and marine management activities required to protect and improve the density and distribution of seagrass.

Background

Western Port is an area of high environmental significance. It has international recognition as a Ramsar site for migratory birds and contains three marine national parks. Seagrass beds are an important aquatic vegetation type, and are widely recognized as iconic and important habitats for fish and other marine biota by the general public (1).

Historically, Western Port had extensive areas of seagrass (primarily *Zostera* sp. (2)). These areas have greatly contracted since the 1970s and the major driver of this decline is thought to be the inputs and resuspension of nutrients and sediments from the catchments feeding the bay (1). Sediments suspended in the water column can block light from reaching the seagrass, and when the sediment settles it can smother and kill the seagrass. High nutrient inputs cause eutrophication that leads to blooms of phytoplankton (suspended microalgae). These blooms cause seagrass dieback through increased turbidity, increased sediment sulphide and epiphytic overgrowth (3). Not much is known about the relative importance of sediments and nutrients, however, or their critical thresholds, if these exist, in controlling seagrass decline in Western Port (1, 3).

There is a spatial pattern of seagrass distribution around Western Port, with higher coverage in the southern and western sections that have the smallest direct input from local rivers and creeks, and the lowest coverage in the eastern section, where there are much greater discharges of nutrient and sediment rich waters from waterways such as the Bunyip and Lang Lang Rivers (2).

For this study, we linked observations with model outputs to derive preliminary estimates of water quality requirements for seagrass. The study had two components:

1. The use of existing maps of seagrass distribution with contemporary mapping and modelling of water quality to investigate whether changes in seagrass distributions are potentially linked with changes in water-quality variables, and whether changes to water-quality have driven seagrass loss or are a result of seagrass loss. Using scenario modelling, we assessed the feasibility of using this approach to derive water-quality guidelines for Western Port.
2. Evaluation of how water quality variables relate to physiological seagrass variables, such as C:N:P ratios and stable isotope ratios of nitrogen and carbon ($\delta^{15}N$ and $\delta^{13}C$) in Western Port, and the feasibility of using these as metrics of seagrass health.

This project forms the first phase of work aimed at deriving water-quality guidelines for Western Port and will feed into the review of the recently expired (July 2011) State Environment Protection Policy Schedule F8, and the upcoming review of Waters of Victoria.

The second phase of this work, part of a three-year Australian Research Council Linkage project led by Monash University and co-funded by Melbourne Water, EPA Victoria and Parks Victoria, will use more up-to-date vegetation maps to investigate threats to seagrass persistence and connectivity and also assess recovery rates.

Methods

Field Work

Field studies took place over eight days in August 2012 and February 2013. Field trips were arranged at times of high tide, to allow access to as many sites as possible.

The most recent vegetation map (from 1999) was used to plan the sampling scheme (4). The intention was to sample up to 90 sites along 30 transects, covering the major areas of Western Port but focussing on the north western (dense seagrass cover) and north eastern (patchy seagrass cover) areas. Three sites were sampled on each transect, covering a depth range from intertidal (0.6-1.2 m) to shallow subtidal (1.5-3.0 m) to deep subtidal (3.3-10.1 m). GIS aerial and LIDAR layers of Western Port were used to plot the sampling sites, ensuring there were sites where seagrass was present and absent. Generally the intertidal site was on the mud flats, the shallow site was on the channel slope, and the deep site was within the channel. All coordinates were taken in datum WGS 1984, Degrees, Minutes and Decimal minutes. At the conclusion of the field work, the sites and the water quality results were plotted with GIS ArcMap 10.

Seventy-six sites were sampled in August 2012 and forty four in February 2013.

Water quality

Samples for water quality analysis were taken using two approaches.

1. A Hydrolab sonde was used to measure temperature, salinity, dissolved oxygen, turbidity, pH, and chlorophyll *a* (fluorescence) at a nominal 1 m depth. Depth profiles at 2 m intervals were collected at 19 deep sites. The sonde was calibrated at Monash University prior to deployment. Photosynthetically Active Radiation (PAR) was measured at the surface and at 1 m depth using a Licor LI1000 logger and SB192 submersible sensor. PAR attenuation was calculated as:

$$\boldsymbol{K_d = (-\ln(I \times 1.34)/I_0)/d},$$

where I = PAR, d = depth, I_0 = PAR at the surface, and 1.34 is the immersion factor for the PAR sensor.

2. Discrete water samples – for total nitrogen (TN), total phosphorus (TP), ammonium (NH_4^+), nitrates and nitrites (NO_x), filterable reactive phosphorous (FRP) and Total Suspended Solids (TSS) – were collected from near-surface waters at all 76 sites. TSS samples were collected in 1 L plastic bottles and stored on ice. Nutrient samples were filtered through 0.4 µm membrane filters, stored on ice and then frozen. Analysis was performed at the Monash University Water Studies Centre Analytical Laboratory using standard methods.

Seagrass Collection

Seagrass was collected by dragging an anchor within 2-3 m of the boat at each site; samples were stored in plastic bags on ice.

Sediment sources

Sediment was collected from the anchor from at least one site per transect and stored on ice. TSS samples were collected by forcing as much water as possible through ashed glass fibre filters. The filters were then wrapped in aluminium foil, stored on ice and frozen at the end of each day.

Stable isotope analysis

The sources of suspended sediment (terrestrial versus re-suspended) and seagrass carbon and nitrogen as well as seagrass condition were investigated using C:N and C:P ratios along with $\delta^{15}N$ and $\delta^{13}C$ values. Low C:N and C:P ratios can indicate an oversupply of bioavailable nitrogen and phosphorus respectively, or low productivity. High ratios indicate nutrient limitation and high productivity. Anthropogenic sources of N often have high $\delta^{15}N$, while internal sources, such as nitrogen fixation, have low $\delta^{15}N$. Terrestrially-derived sediments have low $\delta^{13}C$ while internally produced sediments (e.g. through the breakdown of seagrass and microphytobenthos) have high $\delta^{13}C$.

Seagrass samples were washed in distilled water, separated into leaves, rhizomes and roots where possible, and freeze-dried. Sediment samples were also freeze-dried.

Stable isotope analyses were conducted at Monash University. Seagrass and sediment samples were thawed. The seagrass was gently scraped with a scalpel blade to remove epiphytes. The samples were dried at 60 °C for two days and then pulverized in a mortar and pestle. The material was analysed for $\delta^{15}N$ and $\delta^{13}C$, and TN and total carbon (TC) using a Sercon 20-22 stable isotope ratio mass spectrophotometer.

Benthic vegetation mapping

The 76 sites were surveyed by video in August 2012. A Canon HDV 1080i high-definition video camera in an Ikelite video housing was attached to a frame facing downward with a 50 x 50 cm quadrat attached to the bottom of the frame in view of the camera. The camera was set running on deck at each site, recording site details. The frame and camera were lowered to the seabed at each site using the vessel davit, and left in place for at least 30 seconds.

The field video footage was captured using Adobe Premiere Elements 4.0. This footage was manually classified for dominant substrate and dominant biota (macroalgae and seagrass) within the quadrat using a set of decision rules (5). In each case, the analyst selected from a limited list of alternative descriptors. Substratum was classified as reef or sediment, with percentage of each in quartiles (e.g. reef 51-74%, sediment 26-49%). Sediment was sub-categorised as bare, seagrass, macroalgae or seagrass/macroalgae. Where macroalgae dominated, the three most dominant species and/or mixture of plants were described, along with density (sparse, medium or dense). Seagrass was identified (where possible) to species level, and cover was categorised as dense (thick enough to hide the sediment from view), medium (leaves touching, but sediment visible) or sparse (leaves not touching).

Water-quality modelling

Modelling was undertaken in order to investigate the spatial variation in key drivers of seagrass health and provided simulations of hydrodynamics, locally generated waves and subsequently, sediment transport. Results from these models were used in conjunction with measured data to model light attenuation throughout the bay.

The modelling was used to simulate three yearlong periods: 2011, the most recent full calendar year for which data was available; 1998, the year before the most recent large-scale seagrass survey (2); and 1974, the year of the first comprehensive seagrass survey (2). A shorter simulation of a three month period, June to August 2012, was also undertaken for calibration purposes. The modelling made use of friction maps derived from the seagrass coverage for each modelled year. Seagrass coverage varies considerably between the modelled years with the greatest coverage occurring during 1974. The boundary condition data (wind and sea level) used in the models were available for 1998 and 2011, but not for 1974. Boundary conditions from the 1998 model were therefore used to drive the 1974 simulation.

A bathymetric model grid was developed using combined LIDAR, multibeam and hydrographic chart data. The final bathymetry is a 100 m by 100 m resolution model grid with 583 cells in the x direction and 441 cells in the y direction (Figure 10).

The hydrodynamic model was developed as a 2-dimensional model driven by sea level boundaries on the open coast, wind from two Automatic Weather Stations (AWSs) and river flows from the Ports E2 catchment model (6). The hydrodynamic model was calibrated against sea level data from five sources: three 500 kHz Acoustic Doppler Current Profilers (ADCPs, North West, South West and South); a 1200 kHz ADCP in the South East; and sea level data from the tide gauge at Stony Point (locations shown in Figure 1). The tide gauge at Stony Point was used to fine tune the boundary conditions and consequently should not be considered as a measure of model performance. The model was also calibrated against current speed and direction data collected by the four ADCPs although the data collected from the South ADCP was of dubious quality and was excluded from the calibration. Wind wave modelling was run for a period concurrent with the hydrodynamic modelling periods. The wind wave model was driven by the wind boundary files used for the hydrodynamic model.

Sediment transport modelling used combined results of the hydrodynamic and wave modelling along with friction maps to define bottom shear stress throughout the model domain over the modelled periods. The model makes use of a survey of sediment types throughout Western Port (7). TSS output from the sediment transport model was calibrated against spot measurements taken during field work for the current project. A relationship between TSS and light attenuation (K_d) was developed using measurements of each quantity from a long term sampling program undertaken by EPA Victoria at three locations (Hastings, Barrallier Island and Corinella) around Western Port. This relationship was used to convert modelled TSS into the fraction of incident light at the surface that reached the sediment.

The model output was interrogated to generate a range of maps of oceanographic quantities relevant to seagrass health which were produced as georectified grids for further analysis. Results are restricted to the inner bay, inside the confluence of the two arms, since this region has the largest areas of seagrass and biggest catchment influence. Modelled quantities include significant wave height (Hs), peak period (Tp), current speed, TSS and light penetration. Grids were produced for each modelled quantity to show the mean and the 95th percentile for each of the three years (1974, 1998 and 2011).

Figure 1. Deployment locations of the ADCPs and the tide gauge at Stony Point (image source: Google Earth)

Vegetation mapping

We used vegetation maps from 1974 (15) and 1999 (2) and an updated version of the 1999 vegetation map created in 2009 (from the Department of Primary Industries, Fisheries Research Branch, Habitat Ecology Section).

Historical water quality

Detailed water quality data from 1973-75 (8) was obtained from DEPI, Queenscliff. Water quality in August in these three years was compared with water quality data collected in August 2012. The sampling sites were not the same across the two sampling periods, but each covered a variety of sites around the bay (Figure 2). Many of the sites sampled in 2012 were in areas with high terrestrial inputs and so we would expect this data to overestimate mean nutrient concentrations compared with the 1970s data.

EPA Victoria provided long term monitoring data (1984 to the present) from the three Western Port sites in their Marine Fixed Site Network – Hastings, Barrallier Island and Corinella.

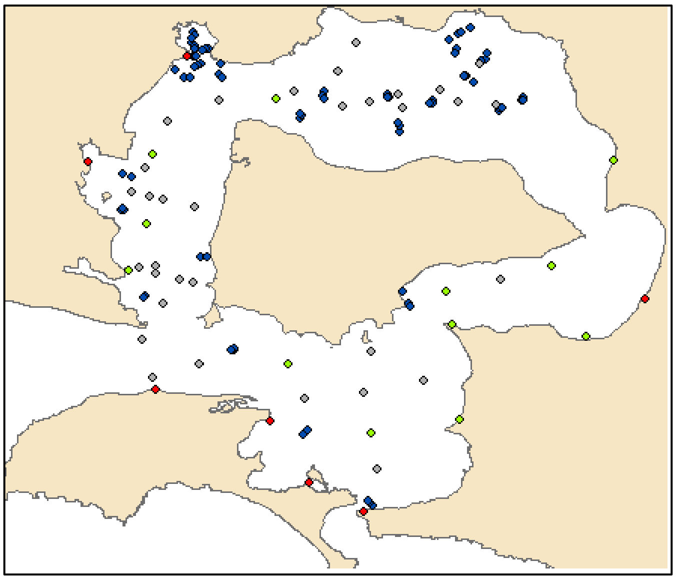

Figure 2. Western Port nutrient sampling sites: grey 1973; green 1974; red 1975; blue 2012.

Seagrass distribution modelling

Seagrass distributions for 1974, 1998 and 2011 were modelled using the computer program Maxent 3.3.3k (9, 10). This program uses maximum entropy density estimation to produce a probability distribution based on location characteristics. For this initial modelling, all locations with seagrass (whether or not they also contained macroalgae) were combined into a single class called 'seagrass' and all locations with macroalgae, but no seagrass, into a second class called 'macroalgae'. The program was 'trained' using the 1999 habitat map (2) and the 1998 models for % light at the sediment surface, mean TSS, % time above threshold, mean wave height and mean wave periodicity, and the depth data.

This trained habitat model was then run using the outputs from 1974 and 2011 hydrodynamic, sediment transport and light penetration models.

Statistics

Water quality parameters measured in August of 1973, 1974 and 1975 were compared with those collected in August 2012 by Analysis of Variance, on log-transformed data and where a significant difference was found we used Tukey's post-hoc test for pairwise comparisons. ANOVAs were also used to compare the mean stable isotope ratios, C:N and C:P of seagrass, sediment and TSS samples collected in August 2012 with those collected in February 2013.

Results

Nutrients

There were clear differences between the two main areas that were sampled: the north-west of the bay (Watsons Inlet) and the north-east of the bay. Water column chlorophyll *a* was low in Watsons Inlet in both August 2012 and February 2013 when compared to the rest of the bay (Figure 3). TSS was also much lower in Watsons Inlet, while NO_x and FRP were much higher (Figure 3). TN and TP showed no trend (Figure 3).

Mean nutrient concentrations in August 2012 were compared to mean nutrient concentrations from August of 1973-1975 (Figure 4). Analysis of Variance indicated a significant difference between years for FRP, TP and NO_x ($p<0.05$) but not for NH_3. Mean TP and NO_x were significantly higher in 2012 than 1973, ($p<0.05$) but there was no difference in FRP. There was no statistically significant difference between 2012 and 1974 or 1975 for any nutrient.

Since 1984, when the Victorian EPA began a regular monitoring program, the trend in nutrient concentrations has been flat, while the TSS concentration has been relatively steady with a possible slight decline after 2000 (Figure 5).

We concluded that the minor differences in concentrations (plus the low chlorophyll *a* in the water column) before and after the seagrass loss indicate that nutrients are unlikely to have contributed much to seagrass decline through the usual eutrophication mechanism (3). Therefore, no further nutrient data were collected during the February 2013 sampling and we focused the rest of the study on light and other physical factors, such as currents and wave height.

Stable isotope ratios and C:N:P

Sediment

Mean sediment isotope ratios were similar over the two seasons. Mean $\delta^{13}C$ was -17.5 ± 1.8 (S.D.) ‰ and mean $\delta^{15}N$ was 4.0 ± 1.4 ‰. $\delta^{13}C$ ranged between -14 and -25 ‰, with the lowest values occurring near creek mouths in the northern section of Western Port (Figure 6) which is likely to be because of terrestrial inputs which tend to have low $\delta^{13}C$. $\delta^{15}N$ ranged between -1 (indicating high rates of nitrogen fixation) and 9 ‰ (indicating anthropogenic inputs), but there was no clear spatial trend (Figure 7).

Total Suspended Solids

The $\delta^{13}C$ of suspended sediments was between -23.8 and -28.8 ‰, much lower than both the sediment and seagrass $\delta^{13}C$ (Figure 6 and Table 1), indicating a terrestrial source of this material at the time of collection (August 2013).

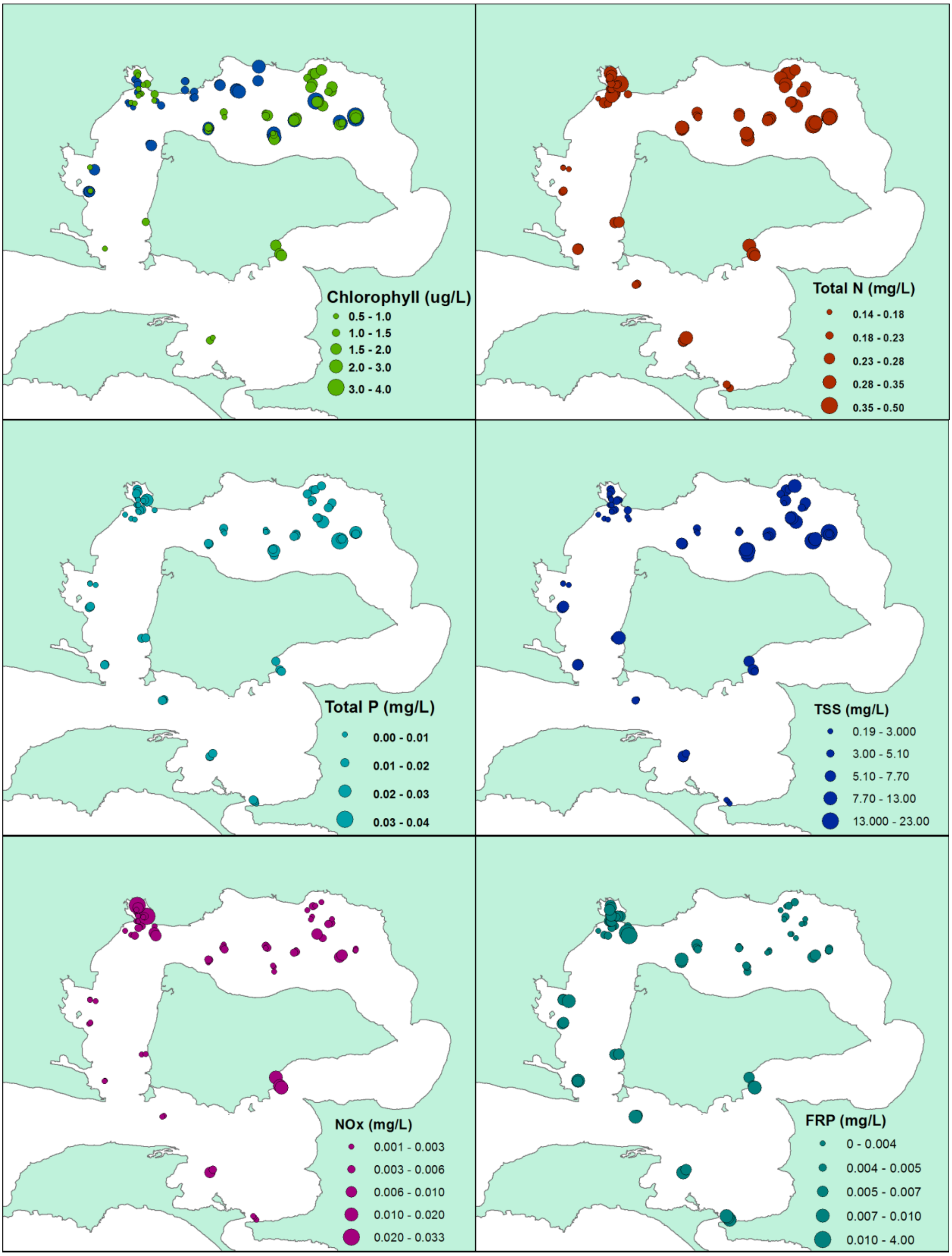

Figure 3. Chlorophyll *a* fluorescence, total N, total P, NO_x, FRP and TSS. All data were collected in August 2012, apart from blue chlorophyll data, which was collected in February 2013.

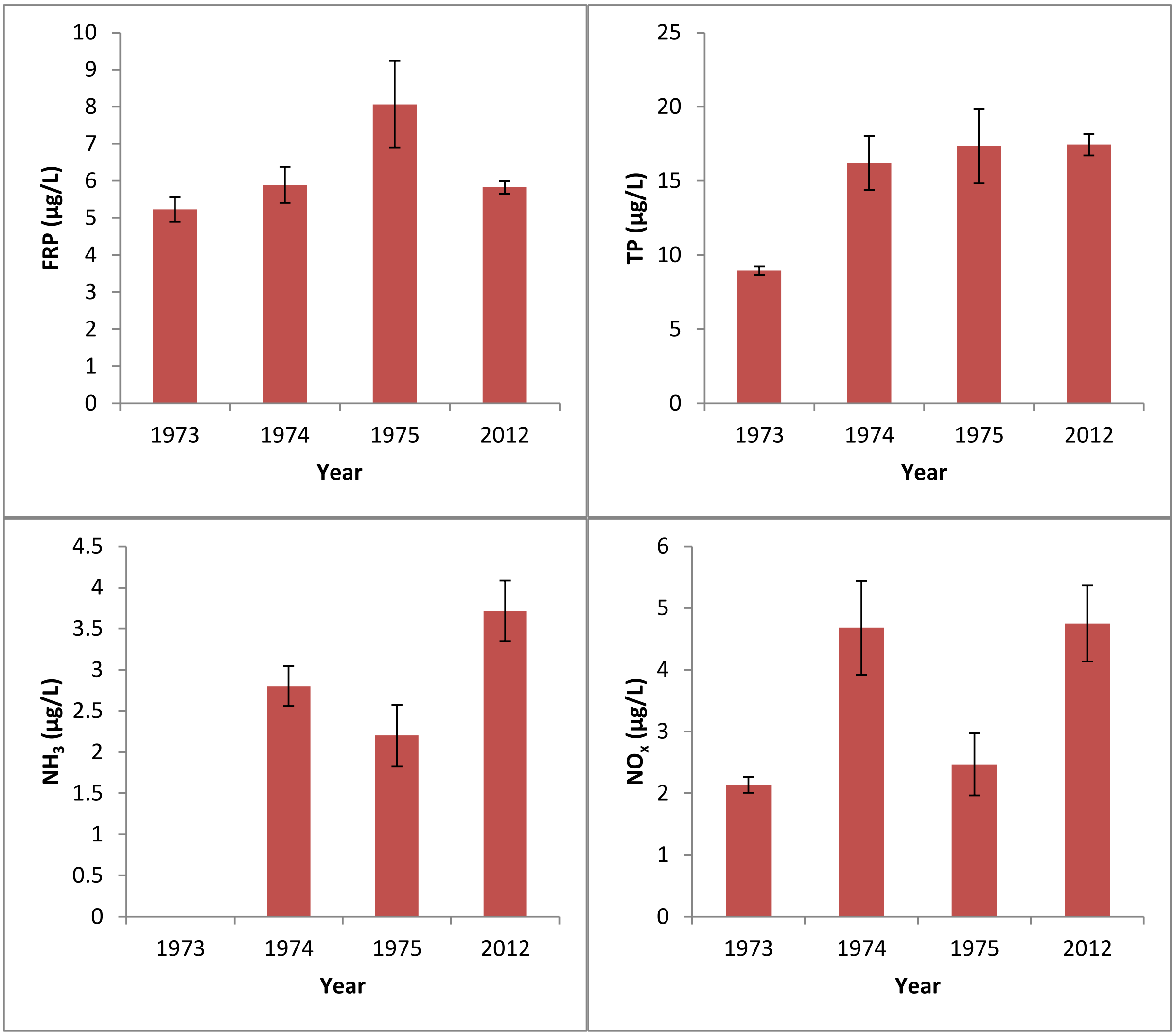

Figure 4. Mean nutrient concentrations (and standard deviation) in Western Port measured in August in the 1970s compared to 2012. Note: there were no NH_3 data in 1973.

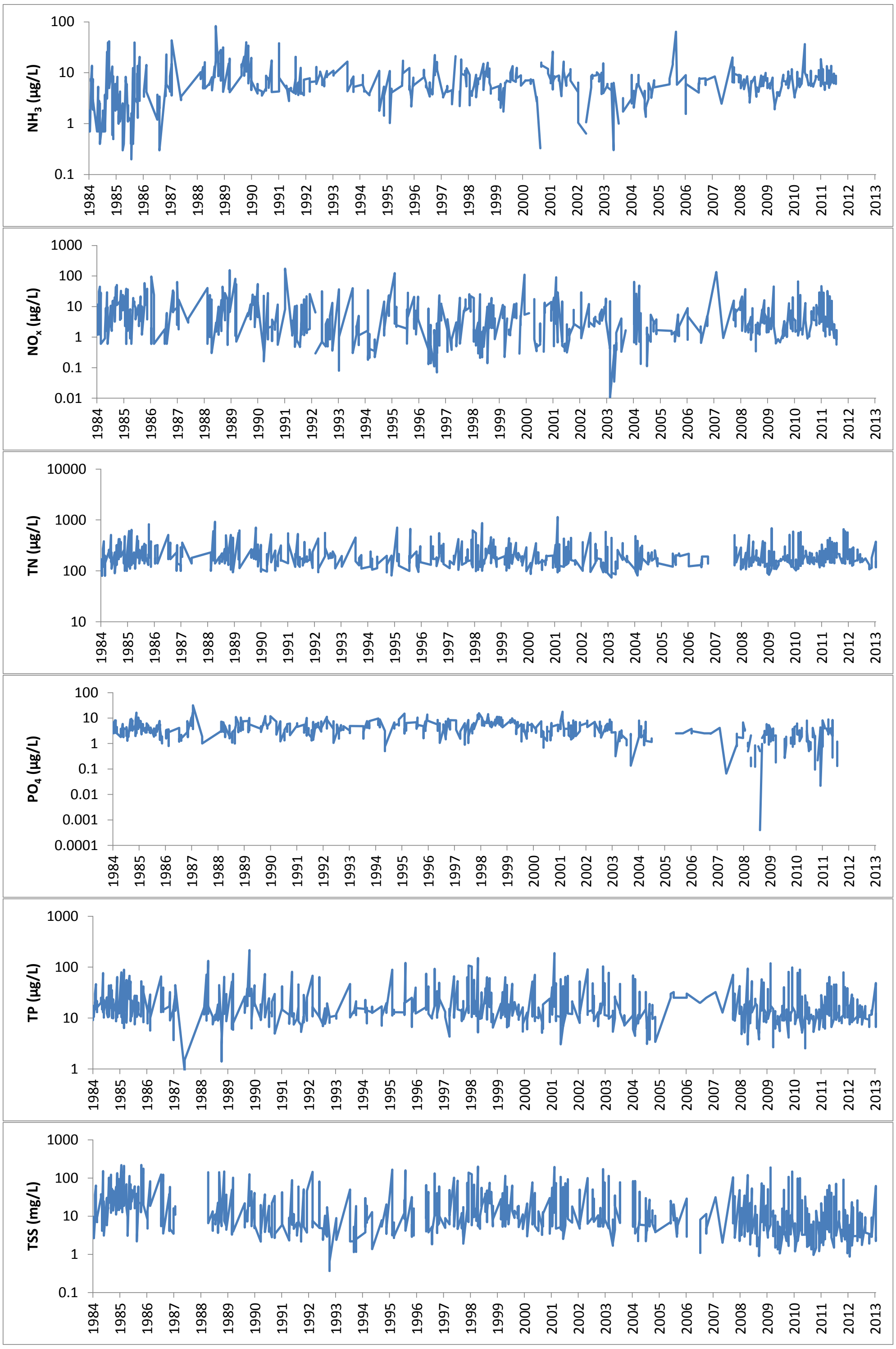

Figure 5. Nutrient and TSS concentrations since 1984 averaged over three sites in Western Port. Data are from EPA Victoria's ongoing Marine Fixed Site Network monitoring program.

Seagrass

There was little difference in $\delta^{15}N$ and $\delta^{13}C$ between seagrass leaves and rhizomes so we present leaf data only.

Seagrass leaf $\delta^{13}C$ (mean -13.7 ± 1.5 ‰) had no evident spatial pattern (Figure 6), but was significantly higher in February (-12.2 ± 1.3 ‰) than in August (-13.8 ± 1.6 ‰, Table 1). This indicates that the seagrass was, as we would expect, more productive in the summer than in the winter.

There was a significant drop in seagrass $\delta^{15}N$ between August (mean 7.0 ± 1.7 ‰) and February (5.3 ± 1.2 ‰, Table 1). During winter, $\delta^{15}N$ values were generally higher in Watsons Inlet and the northern sections of the bay (Figure 7) reflecting known inputs of 'heavy' nitrogen from anthropogenic sources such as agriculture and treated sewerage (11). The drop in $\delta^{15}N$ in summer can be attributed to an increase in nitrogen fixation in seagrass beds combined with a drop in catchment inputs over this period (11).

There was little spatial variability in seagrass C:P and C:N ratios (Figure 8) but there was a distinct increase in both of these ratios between winter and summer. Mean C:P (mass:mass) was 185 ± 36 in August and 242 ± 49 in February, and mean C:N was 14 ± 2 in August and 22 ± 2 in February; these differences were highly statistically significant (Table 1). This change shows the effect of rapid biomass (carbohydrate) accumulation in summer, compared to low growth with N and P accumulation in winter. There was a negative correlation between C:N and water column NO_x (Figure 9) and therefore C:N may be useful as a predictor of nutrient limitation or oversupply.

Table 1. Stable isotopes and mass:mass ratios of carbon and nitrogen. P-values are from t-tests comparing August and February.

Substrate	Month	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	C:N	C:P
Sediment	August	-17.6 ± 2.1	3.8 ± 1.6	9.7 ± 5.5	NA
	February	-17.3 ± 0.9	4.4 ± 0.6	8.4 ± 0.9	NA
	P-value	0.692	0.240	0.437	NA
TSS	August	-26.4 ± 1.2	4.7 ± 0.3	8.6 ± 0.7	NA
Leaf	August	-13.8 ± 1.6	7.0 ± 1.7	14 ± 2	185 ± 36
	February	-12.2 ± 1.3	5.3 ± 1.2	22 ± 2	242 ± 49
	P-value	<0.001	<0.001	<0.001	<0.001

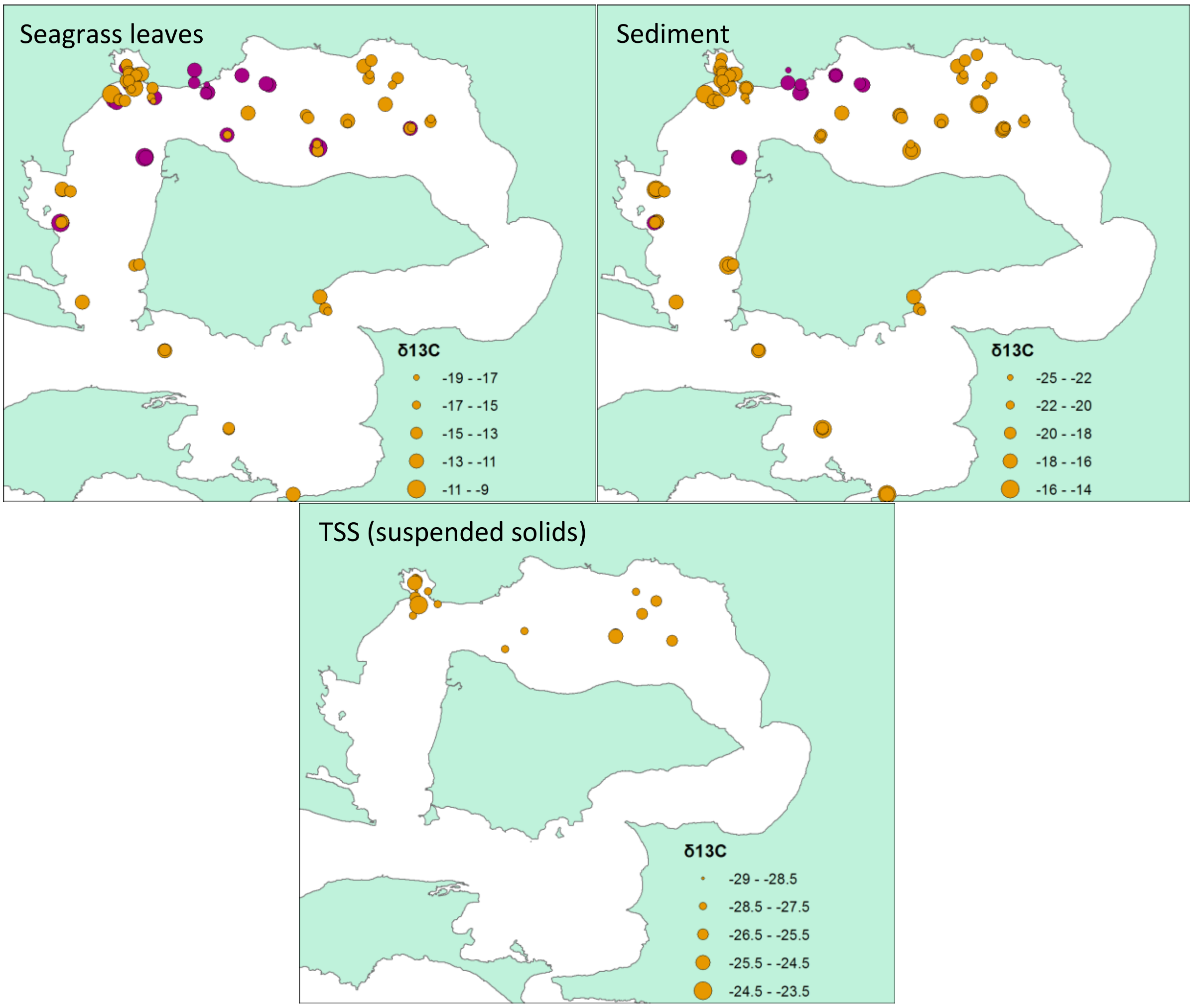

Figure 6. δ^{13}C (‰) of seagrass leaves, sediment and TSS. Orange dots are from August 2012 and purple dots from February 2013 (no TSS data for February 2013). Note the different scales.

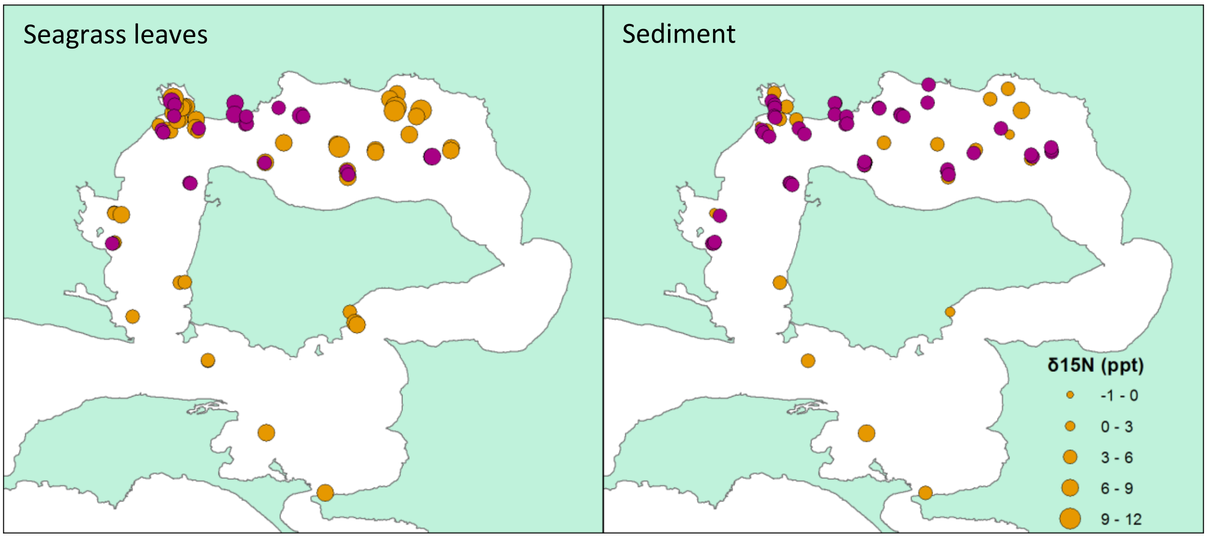

Figure 7. δ^{15}N (‰) of seagrass leaves and sediment. Orange dots are from August 2012 and purple dots from February 2013.

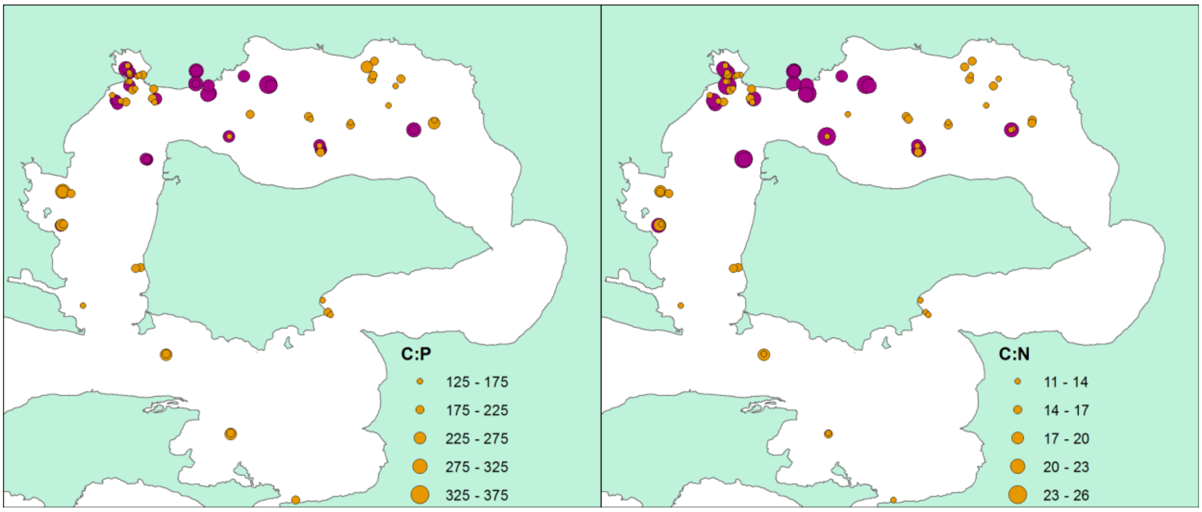

Figure 8. Seagrass leaf C:P and C:N (mass:mass). Orange dots show August 2012 data and purple dots show February 2013 data.

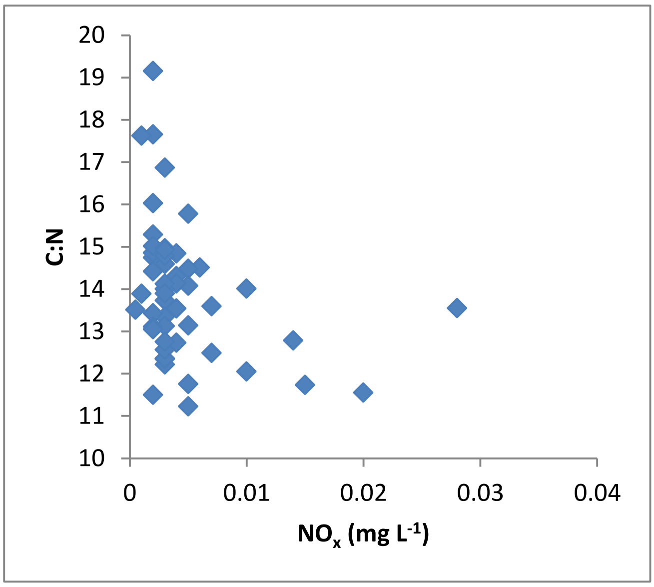

Figure 9. Water column NO_x plotted against C:N. As expected, C:N is low at higher NO_x concentrations.

Western Port model outputs

Overall, there was good agreement between the model outputs and the observed sea level data, current speed and direction, and wave height and the model performs.

Modelled TSS was calibrated against the TSS data collected in August 2012. An example of the calibration is shown in Figure 10. The trends in TSS variation are generally well represented by the model, but the occasional outlier is evident (e.g. sample 1 in Figure 10).

The modelling showed that the greatest inter-annual variability in model results was due to the differences in seagrass coverage with areas of seagrass representing areas of high friction. In the simulation of 1974, wave height (Figure 12 and Figure 13) and current speed (Figure 16 and Figure 17) were attenuated over the large seagrass patches and TSS values were consequently reduced (Figure 19 and Figure 20). Lighter winds in 2011 also produced reduced wave height and consequently lower overall TSS levels than 1998.

The biggest increases in the time above threshold for erosion (Figure 18) and TSS (Figure 19 and Figure 20) were in the eastern and north-eastern sections of the bay, where seagrass loss since the 1970s has been the most severe. There was no major change in erosion and or TSS in the north-western section of the bay, where seagrass coverage has remained stable through time.

There were no major differences in the modelled light at the sediment surface between the three periods (Figure 21).

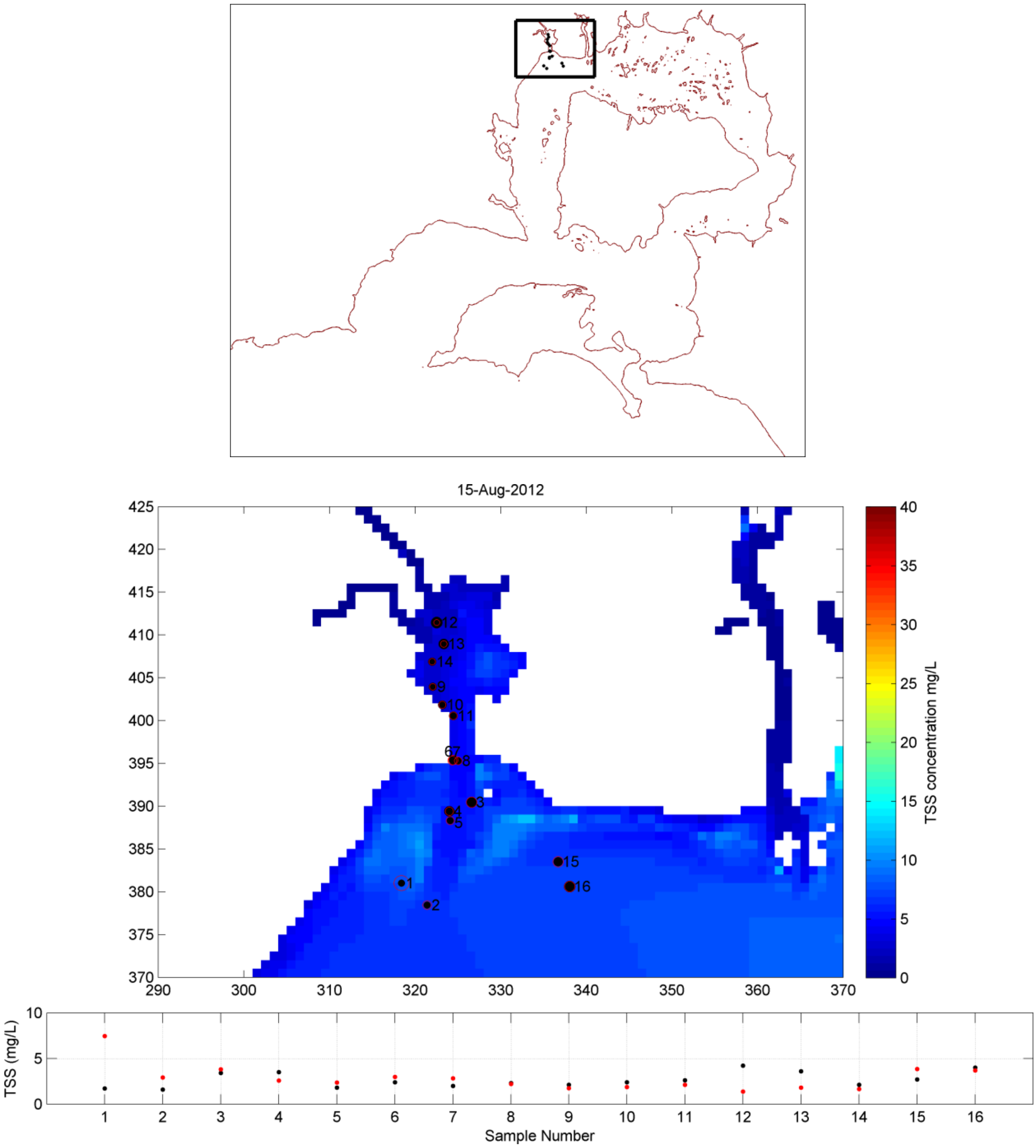

Figure 10. Top: broad scale location of TSS samples taken on the 15th Aug 2012. Middle: fine scale location of samples with black filled circles indicating measured quantity and red circle indicating the modelled value. Bottom: Comparison of measured (black) versus modelled (red) TSS values.

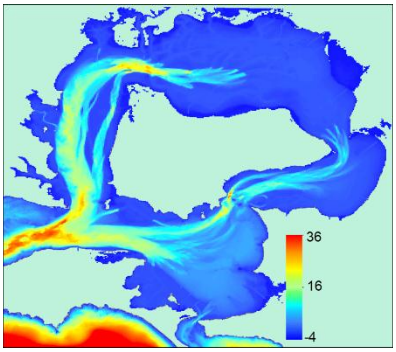

Figure 11.Western Port bathymetry. Depth in metres.

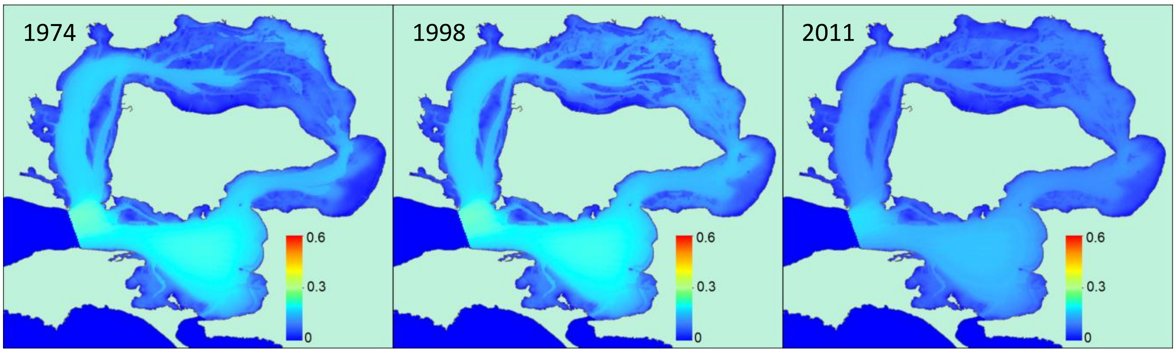

Figure 12. Mean wave height (m).

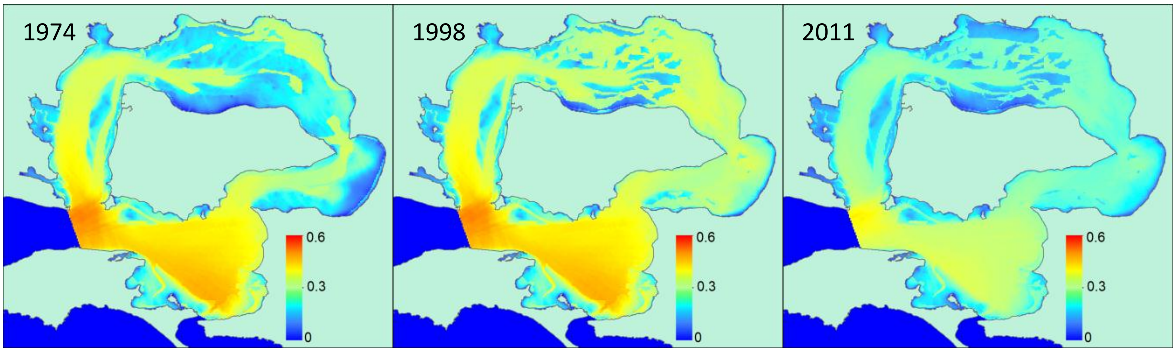

Figure 13. 95% wave height (m).

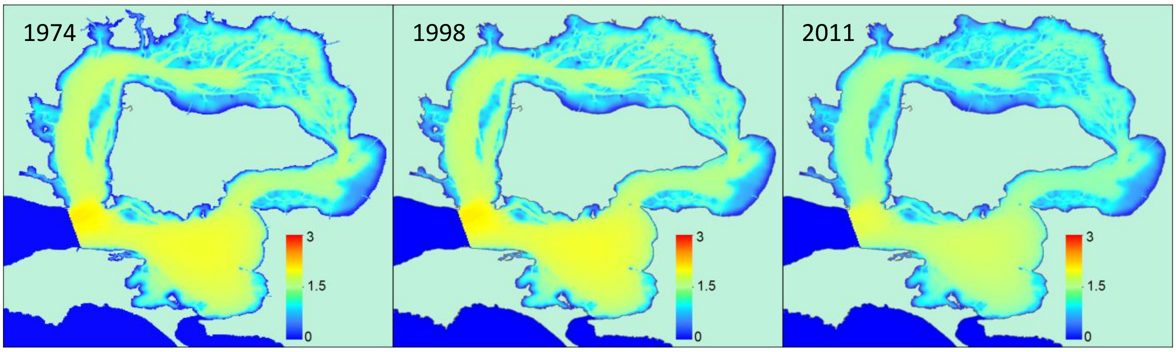

Figure 14. Mean wave period (s).

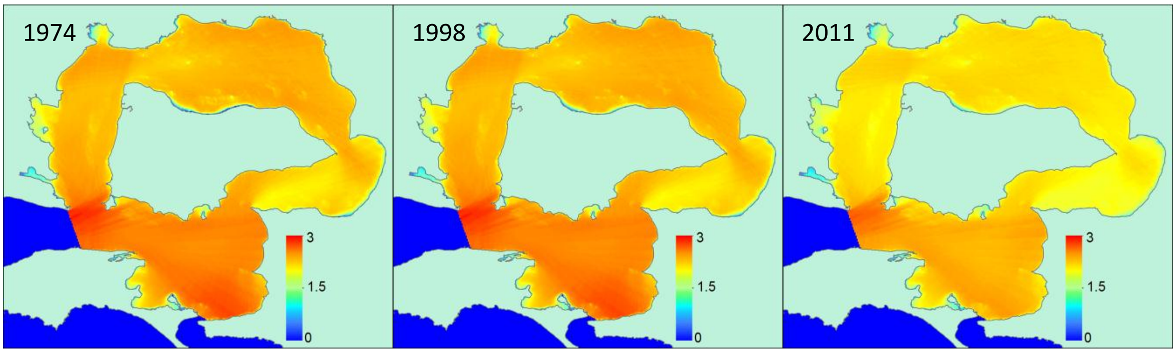

Figure 15. 95% wave period (s).

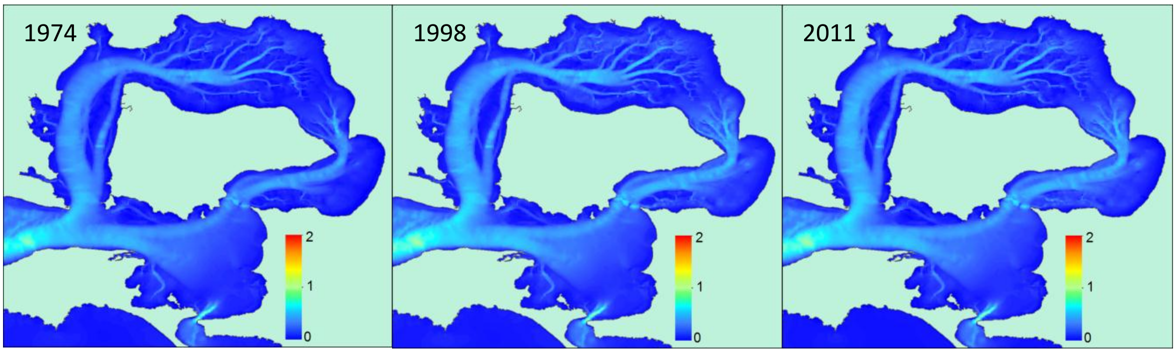

Figure 16. Mean current speed (m s^{-1}).

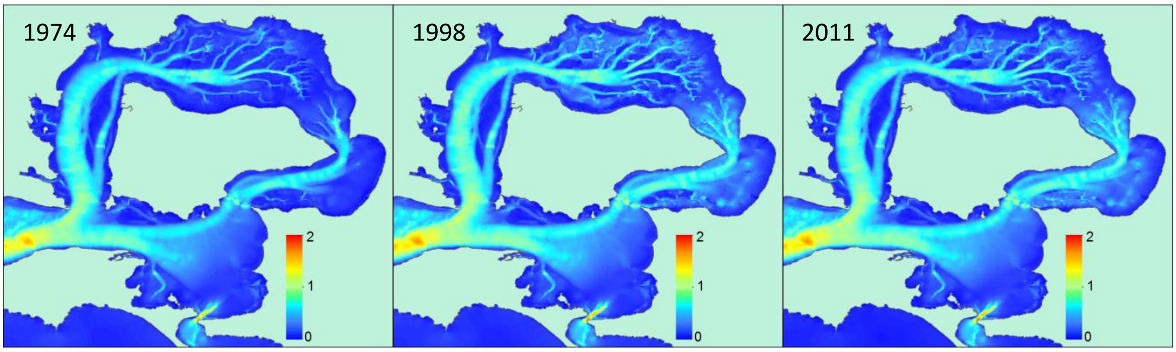

Figure 17. 95% current speed (m s^{-1}).

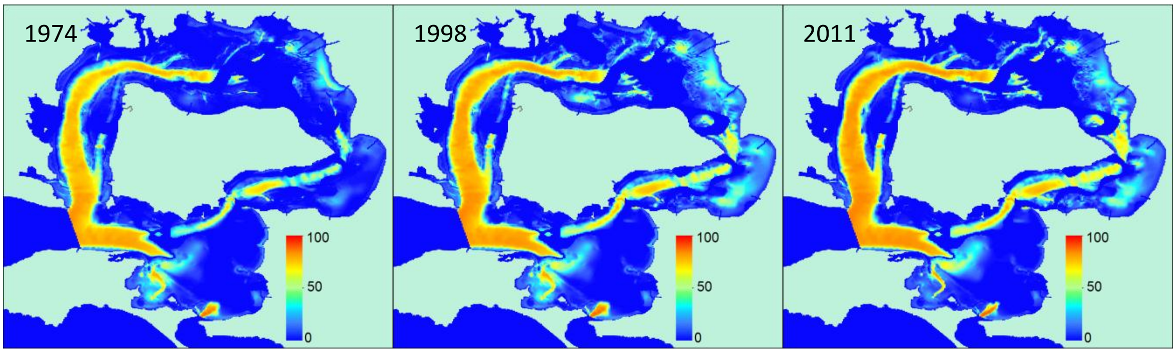

Figure 18. % time above threshold for erosion.

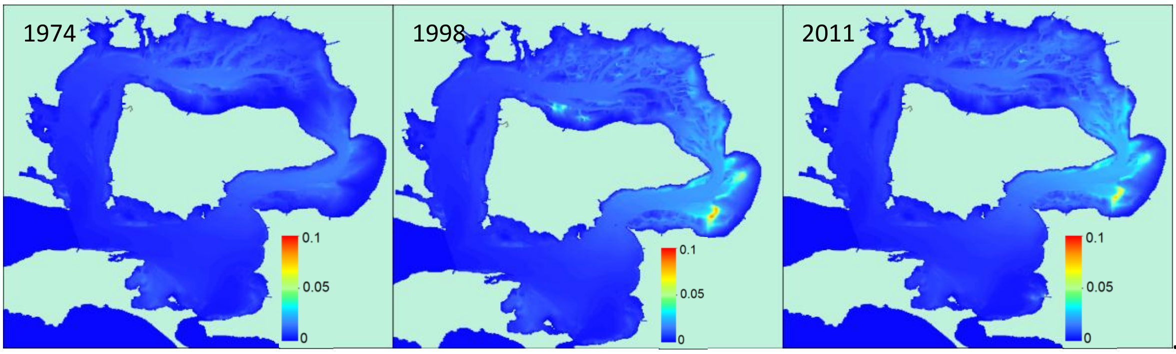

Figure 19. Mean TSS (mg L^{-1}).

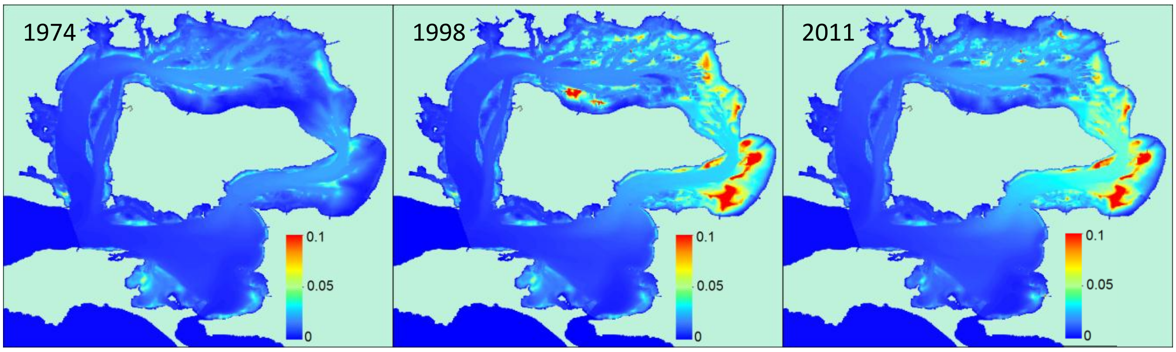

Figure 20. 95% TSS (mg L^{-1}).

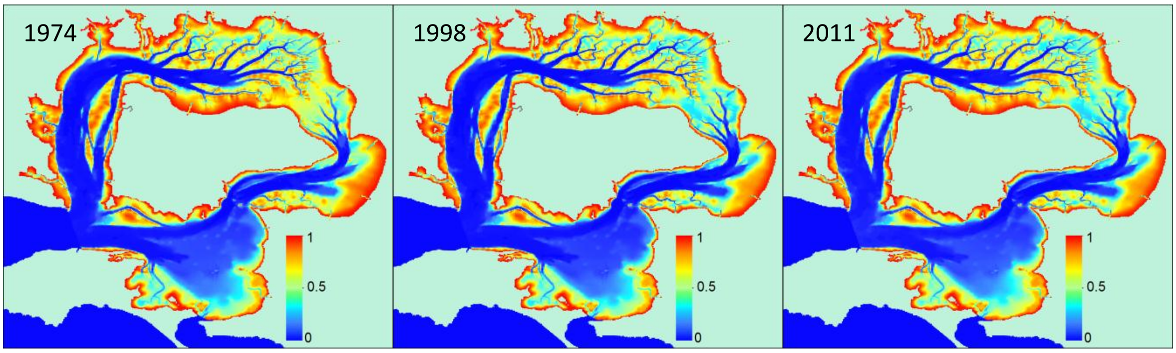

Figure 21. % light at sediment surface.

Maximum entropy habitat distribution maps

The output of the Maxent models for 1998 (using the 1998 hydrodynamic and sediment transport models and the 1999 seagrass distribution) closely matched the distributions observed in 1999, apart from the macroalgae model, which extended the distribution further into the east of the bay than was observed in 1999 (Figure 22).

The models show a large increase in macroalgae cover between 1974 and 1998, and an even larger decrease between 1998 and 2011 (Figure 22).

The models show a large decline in seagrass cover between 1974 and 1998, particularly in the areas north and east of French Island (Figure 23). The model for 2011 shows some recovery of seagrass in these areas, especially in the northern area (Figure 23). This model also shows an increase in the south-east of the bay in 2011 and suggests that the seagrass distribution there is now greater than seen in the 1970s.

The 1974 model (Figure 23) closely replicated the observed seagrass distribution in 1974 (Figure 24).

In 2009, the Department of Primary Industries observed an increase in seagrass in the central north of the bay and a decrease in the northwest and southeast (Figure 24). The distributions predicted by Maxent (Figure 23) match these observations.

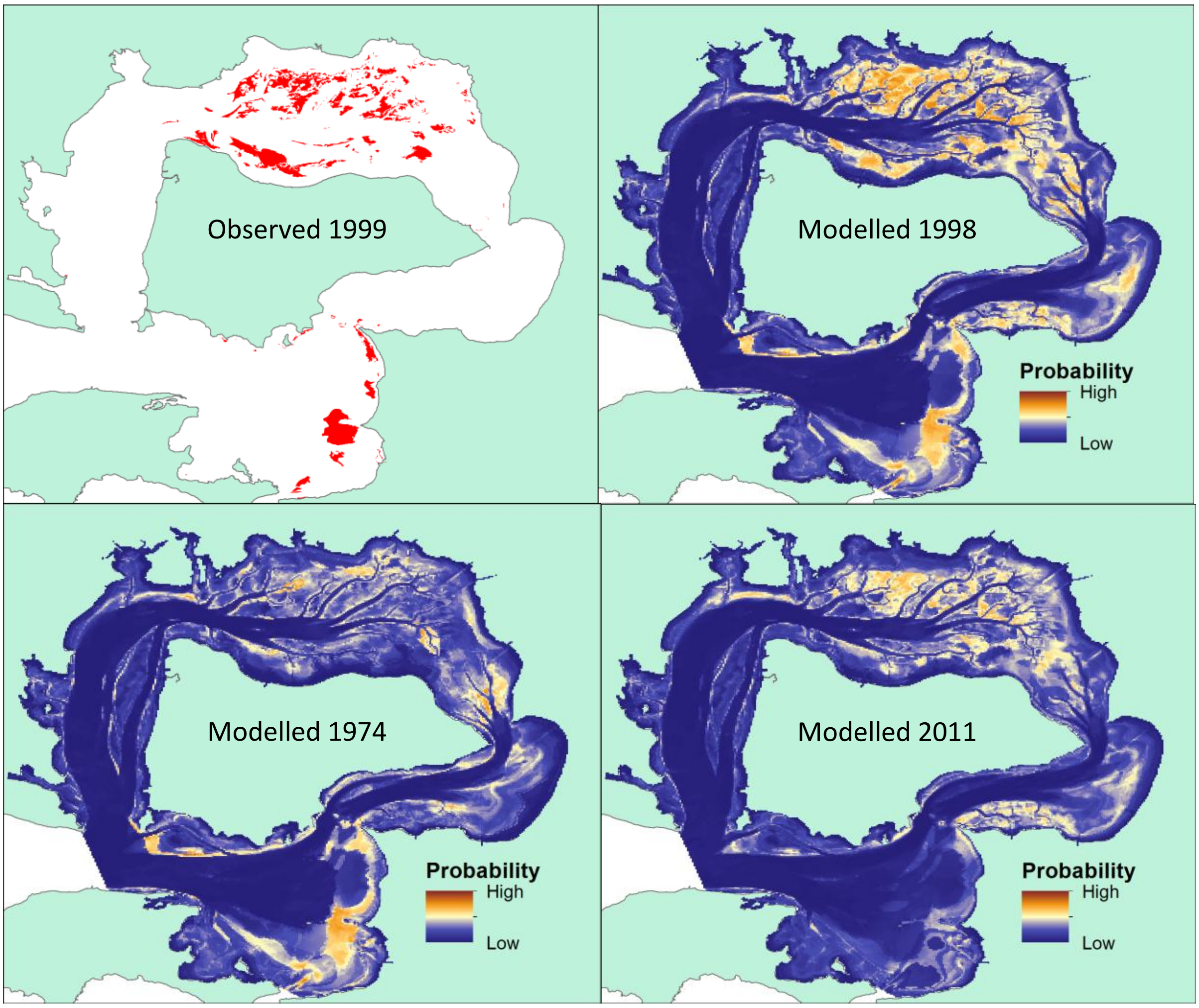

Figure 22. Macroalgae distribution. Observed (1999) and modelled for 1974, 1998 and 2011.

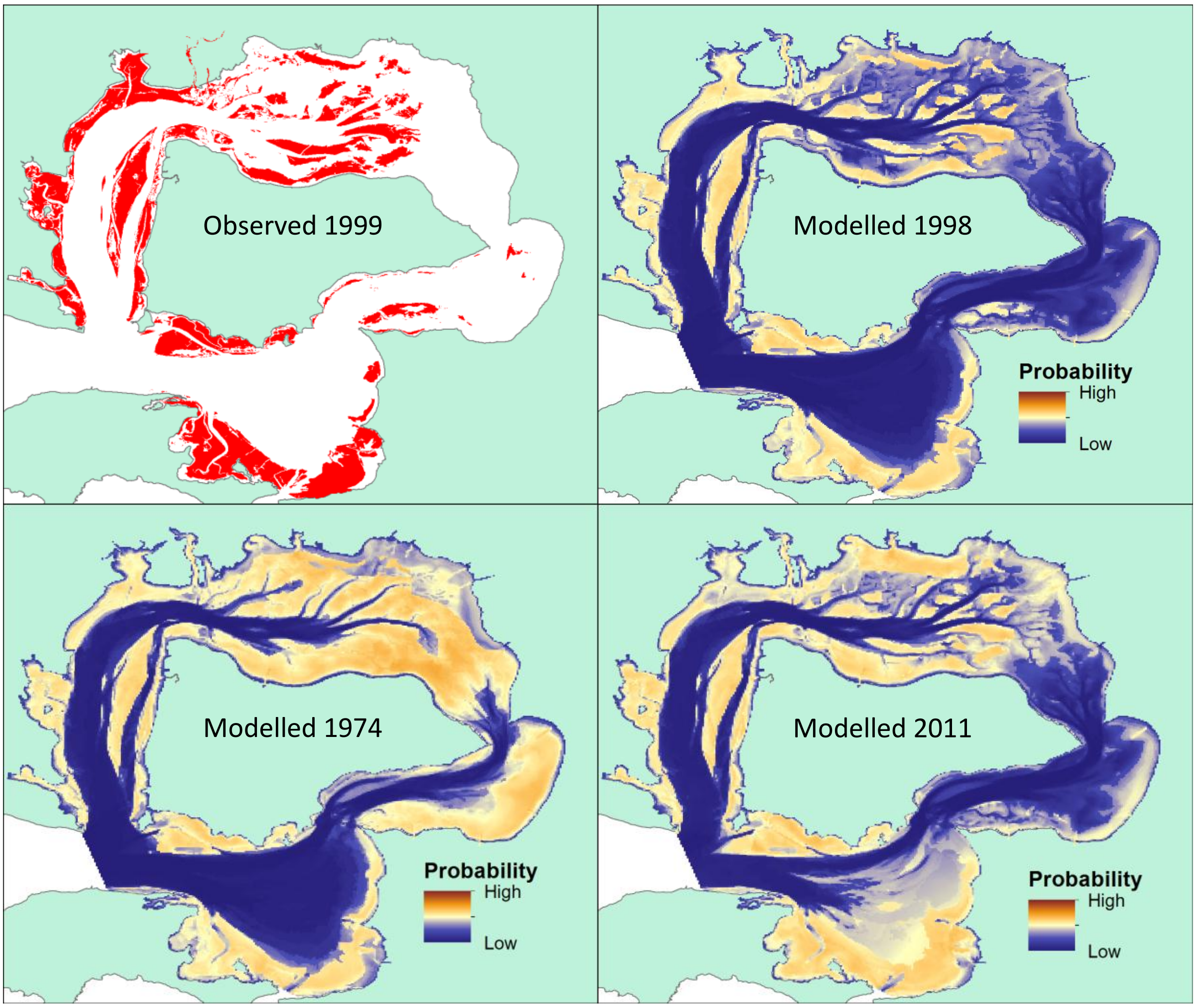

Figure 23. Seagrass distribution. Observed (1999) and modelled for 1974, 1998 and 2011.

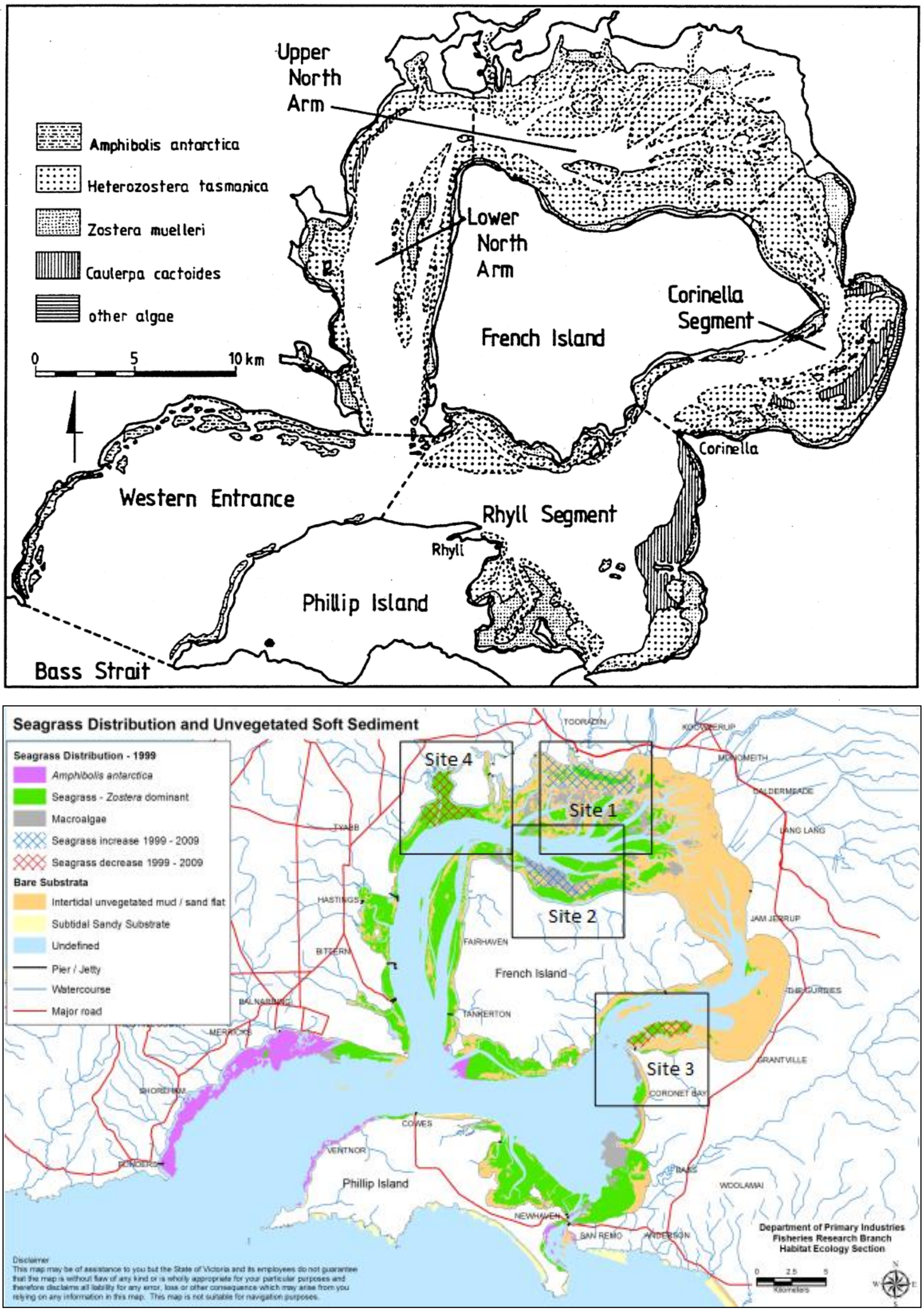

Figure 24. Observed Western Port seagrass distribution in 1974 (top panel, taken from (2)) and 2009 (analysis of aerial photographs, Department of Primary Industries, Fisheries Research Branch, Habitat Ecology Section).

Discussion

Light and total suspended solids as the critical factors limiting seagrass distribution

There is little evidence that eutrophication has had a major impact on seagrass distribution in Western Port. In particular, areas with the highest nutrient concentrations such as Watsons Inlet also have some of the highest seagrass densities. Instead, an increase in suspended particles, and accompanying drop in light availability, seems to be the likely cause of the loss of seagrass, with the highest TSS values being observed in the north east of Western Port where seagrass is sparse or absent. Further support for the hypothesis that light limits seagrass growth comes from the leaf carbon isotope data. As light becomes limiting, seagrass growth slows, which results in less carbon isotope fractionation during CO_2 assimilation. At low rates of photosynthesis, the carbon isotope signature becomes more negative (isotopically lighter) and vice versa (12). This effect was seen in Port Phillip, where the δ^{13}C of seagrass declines with depth before stabilizing at a δ^{13}C of ~-16 to -15 ‰ at ~4 m depth, which we hypothesize represents a threshold rate of photosynthesis at which seagrass can survive (Figure 25 left panel). Within Western Port, we observe that this threshold is reached within the top 1-2 m of water (Figure 25 left panel).

To further explore the hypothesis that light controls the δ^{13}C of seagrass in Western Port, we plotted the proportion of light reaching the sediment surface against the δ^{13}C of sediment, rhizomes and leaves (Figure 25 right panel). Light seems to have no effect on δ^{13}C in the sediment or in the seagrass rhizomes, but there is a correlation between light and δ^{13}C in the seagrass leaves. In this case, the correlation between light field and δ^{13}C is weak compared to previous observations from the literature (12). This is most likely because the light fields are modelled and so may differ significantly from the actual light field experienced by Western Port seagrass. Furthermore, the light field modelled here includes periods of inundation and exposure. Photosynthesis in seagrass is greatly reduced when exposed to the air (13), so ideally only the light field during inundation should be used. This hypothesis and the relationship between δ^{13}C and light availability will be explored as part of an ARC linkage project (LP130100684) to be completed in 2016.

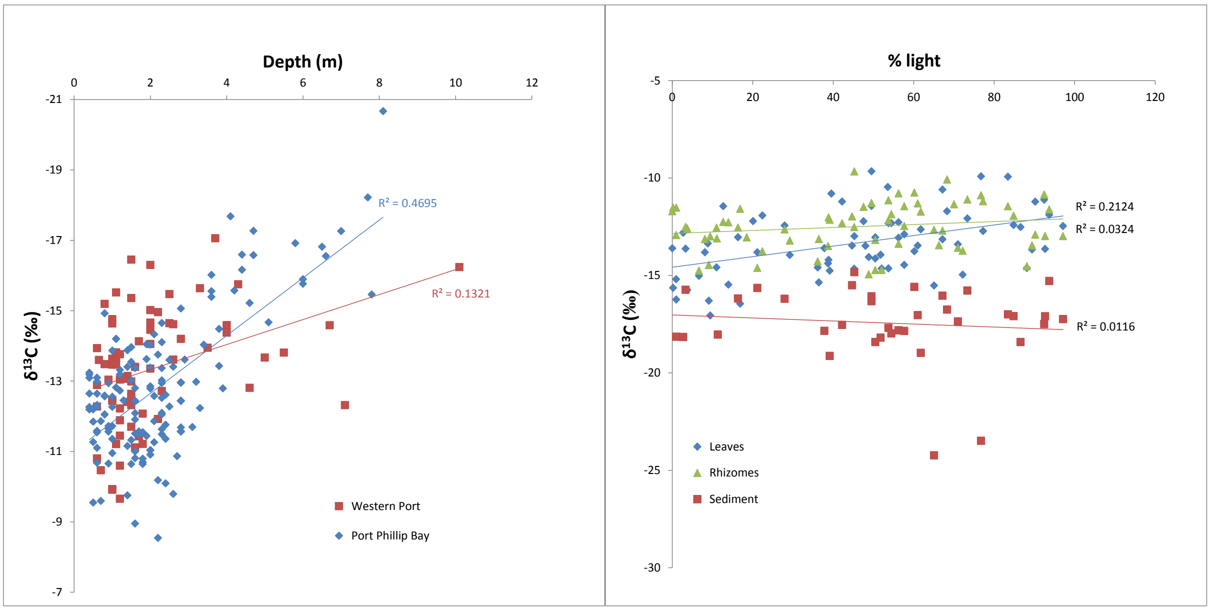

Figure 25. LHS: Seagrass $\delta^{13}C$ of seagrass leaves versus depth; comparison between Port Phillip and Western Port. RHS: $\delta^{13}C$ versus % light in Western Port; comparison between leaves, rhizomes and sediment.

If % surface light field (as currently modelled) is not a useful light metric in the intertidal zone, perhaps TSS is. Our aim was to identify thresholds for key water quality variables, and TSS is relatively easily measured so we considered TSS rather than light per se. If we plot the depth versus mean TSS (modelled 1998) for which dense, medium and sparse seagrass occurred (observed in 1999), some patterns emerge (Figure 26). The occurrence of seagrass decreases as TSS increases. There was a clear difference in the thresholds at which seagrass disappeared for the intertidal (depth < 0) and the subtidal. Within the subtidal, 90% of the dense seagrass occurred where TSS was less than 0.007 mg L^{-1}, with a similar threshold of 0.012 mg L^{-1} for medium seagrass followed by a threshold of 0.019 mg L^{-1} for sparse seagrass. The depth above which 90% of subtidal seagrass occurred was similar for all three densities: 2.6 m for dense seagrass; 2.9 m for medium seagrass and 2.4 m for sparse seagrass. In the intertidal zone, the threshold for 90% of seagrass was similar for all seagrass densities, at ~0.01 mg L^{-1}. The depth below which 90% of seagrass occurred was 1.4 m for sparse seagrass, 1 m for medium seagrass and 1.2 m for dense seagrass. TSS also seems to have a negative effect on the presence of dense seagrass in the high intertidal zone (Figure 26). We have no clear explanation for this, but we hypothesize that the presence of TSS may make seagrass more susceptible to desiccation, possibly through particles being irreversibly stuck onto seagrass leaves once dried.

The above preliminary TSS thresholds for seagrass cover have been mapped for 1998 in Figure 27. These thresholds closely match observed seagrass cover in the southern, western and north-western parts of the bay, but greatly overestimate the seagrass cover in the northern and eastern parts, especially in the intertidal zone. We have two tentative explanations for this: 1. We have limited TSS data for the mudflats in these parts of the bay, and it may simply be that the TSS model underestimates TSS in these areas; 2. Factors other than just TSS may be affecting seagrass in the northern and eastern parts of the bay.

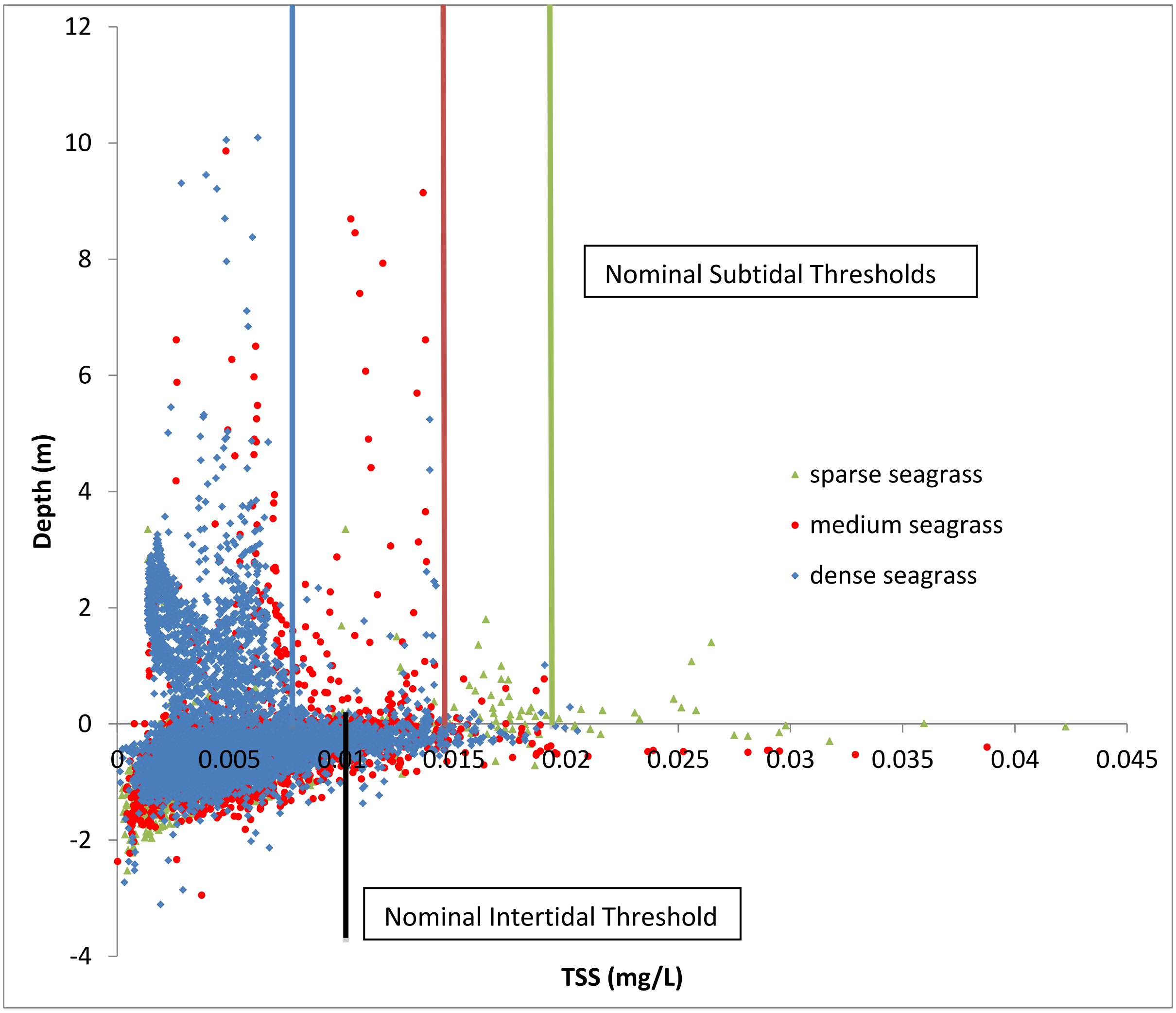

Figure 26. The occurrence of sparse, medium and dense seagrass as a function of TSS and depth. Negative depths represent the intertidal, positive depths subtidal. The data points combine the seagrass distribution observed in 1999 (Figure 23), mean TSS modelled for 1998 (Figure 19) and depth from LIDAR bathymetry (Figure 11).

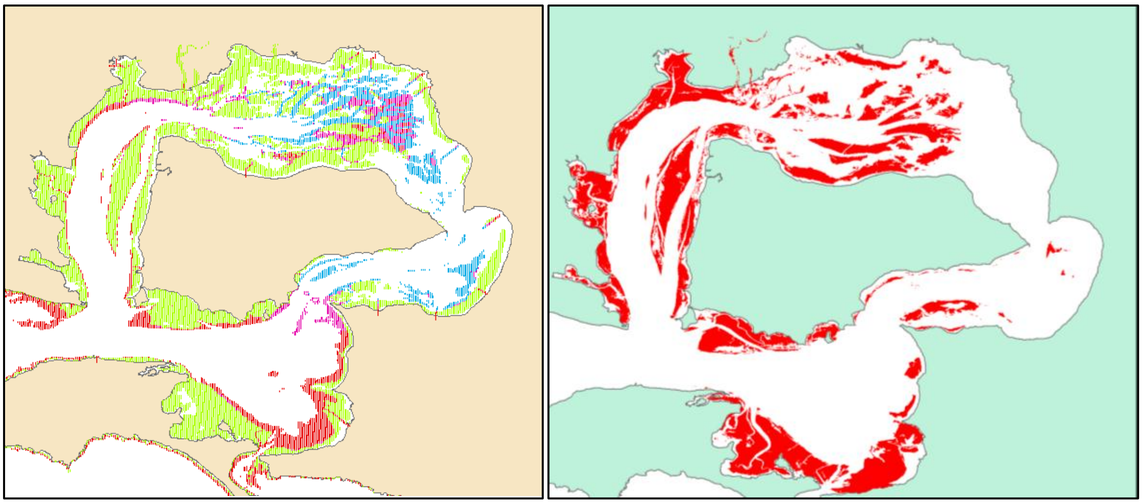

Figure 27. LHS: Potential seagrass distribution based on 1998 modelled TSS and depth. Green is intertidal seagrass, defined as any point between -1.4 and 0 m and less than 0.01 mg L^{-1} TSS (see Figure 26). Red is dense subtidal seagrass (TSS<0.07 mg L^{-1}, depth<2.6 m), pink is medium subtidal seagrass (TSS<0.12 mg L^{-1}, depth<2.9 m) and blue is sparse subtidal seagrass (TSS<0.19 mg L^{-1}, depth<2.4 m). RHS: Observed seagrass cover in 1999.

The importance of TSS in controlling seagrass distribution raises the question of its source. The models show that changes in seagrass distribution have a strong effect on the frictional velocity maps (and vice versa), which suggests that resuspension of sediment within Western Port is the most likely source of the increase in TSS. This agrees with the previous work by CSIRO that found that most of the TSS was derived from wave and wind resuspension of sediment (14). Our $\delta^{13}C$ results, however, conflict with our hypothesis of internal resuspension. Sediment samples taken from the bottom had a $\delta^{13}C$ in the range of -14 to -25 ‰ (Figure 6), consistent with organic carbon being derived from seagrass and phytoplankton. This highlights the fact that although most of the sediment is likely to have originally been derived from terrestrial sources, it rapidly takes on a signature reflecting a mixture of phytoplankton, microphytobenthos and seagrass. The sampled suspended sediment $\delta^{13}C$, however, fell in the range of -25 to -29 ‰ (Figure 6), strongly suggesting this TSS was terrestrial (including mangrove), which typically has a carbon isotope signature of ~-28 ‰. The suspended sediment samples were taken during August, which coincided with a period of maximum freshwater input and so this one off sampling may not be representative. The other uncertainty is the time taken for terrestrially derived sediment to take on a marine isotopic signature. We hypothesise this will take weeks to months which is very short compared to the likely residence time of sediment in Western Port. Further work needs to be undertaken to establish the source of TSS and this will be undertaken as part of the ARC linkage project to follow on from this study (LP130100684).

Indicators of nutrient exposure

Seagrass $\delta^{15}N$ during August was heavier in the northern section of Western Port, particularly around known point sources of isotopically enriched (heavier), anthropogenic nitrogen such as Watsons Inlet (Figure 7). While nitrogen inputs to Western Port do not seem to be an immediate threat to seagrass health, we can use $\delta^{15}N$ of seagrass leaves to monitor their exposure to catchment derived nitrogen. There was a drop in $\delta^{15}N$ between August 2012 and February 2013. Nitrogen fixation in the summer will lead to an increase in atmospherically derived nitrogen in the system and this is probably what is showing up in the seagrass (15). Like with the stable isotopes, the ratio of carbon to nitrogen and phosphorus showed no clear spatial trends, but showed clear seasonal trends (Figure 8). In summer, rapidly growing seagrass will use up internal stores of nitrogen and phosphorus, leading to an increase in C:N and C:P, as was seen in this study. When dissolved nitrogen is available, we would expect the C:N ratio to drop, which seems to be the case (Figure 9).

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